

# **EXHIBIT 94**

BARR

## **DRAFT Conceptual Modeling of PFOA Fate and Transport: North Bennington, Vermont**

Prepared for  
Saint-Gobain Performance Plastics

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# DRAFT Conceptual Modeling of PFOA Fate and Transport: North Bennington, Vermont

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## Acronyms and Abbreviations

<b>Acronym</b>	<b>Description</b>
AHC analysis	Agglomerative Hierarchical Cluster Analysis
DWHA	Drinking water health advisory. As of March 7, 2016, Vermont Department of Health's drinking water health advisory for PFOA was set at 20 parts per trillion (VTDH, 2016a). As of June 22, 2016, Vermont Department of Health's drinking water health advisory for the sum of PFOA and PFOS is 20 parts per trillion (VTDH, 2016b).
MVA	Multivariate analysis
ng/g	nanogram per gram, typical reporting units for analyses of PFOA in soil, equivalent to ppb
ng/L	nanogram per liter, typical reporting units for analyses of PFOA in water, equivalent to ppt for water samples with a density of 1 gram per centiliter (i.e., low total dissolved solids)
PCA	Principal Component Analysis
PF compounds	Perfluorinated compounds, aka PFCs. Used in some cases as a synonym for PFAS
PFAS	Per- and polyfluoroalkyl substances
PFBS	Perfluorobutanesulfonic acid
PFHxA	Perfluorohexanoic acid (C6)
PFHxS	Perfluorohexanesulfonic acid
PFHpA	Perfluoroheptanoic acid (C7)
PFOA	Perfluorooctanoic acid (C8)
PFOS	Perfluorooctanesulfonic acid
PFTeDA	Perfluorotetradecanoic acid (C14)
PFTrDA	Perfluorotridecanoic acid (C13)
PFUnDA	Perfluoroundecanoic acid (C11)
POET	Point-of-Entry Treatment
ppb	Parts per billion
ppt	Parts per trillion
SSV	Soil Screening Value
VTDEC	Vermont Department of Environmental Conservation
VTDH	Vermont Department of Health
WWTP	Wastewater Treatment Plant

## 1.0 Executive Summary

For purposes of this report, numerical models were developed to evaluate and simulate potential transport mechanisms of perfluorooctanoic acid (PFOA) released by historical air emissions at the former Chemfab facilities in Bennington and North Bennington, Vermont that may have resulted in the presence of PFOA in soil and groundwater in the vicinity of these facilities. The results of the simulations have been compared to measured PFOA concentrations in soil and groundwater within the study area to identify areas where exceedances of regulatory standards may potentially be associated with historical air emissions from the former Chemfab facilities.

North Bennington is located in the Northeastern Appalachians groundwater region; an area characterized by rolling topography that primarily reflects the weathered bedrock surface and landforms created by glaciers and rivers mantling the bedrock. Stratified drift units, chiefly sand and gravel, follow the larger valleys such as those of the Walloomsac River and its tributaries. Recharge to the stratified drift units takes place from direct infiltration of precipitation, and discharge from bedrock. Discharge from the stratified drift takes place by pumping of wells, evapotranspiration, and discharge to the larger rivers when their stage is at or below typical levels.

The bedrock consists of folded and faulted, metamorphosed sedimentary rocks with low primary porosity. Water is conducted through the bedrock in secondary porosity (fractures and, in some rock types, solution features). Most bedrock wells in the region are up to 200 to 400 feet deep. Recharge to the bedrock is controlled primarily by the permeability and thickness of the overlying till. Discharge from the bedrock is to wells and to the stratified drift in the large valleys.

Potential PFOA sources in the study area that have been identified to date include:

- The former Chemfab facility on Water Street in North Bennington. This facility operated between 1978 and 2002, during which time PFOA emissions may have occurred.
- The former Chemfab facility on Northside Drive in Bennington. This facility operated between 1969 and 1978, during which time PFOA emissions may have occurred.
- Sludge from the Bennington Wastewater Treatment Plant (WWTP) that was land-applied on agricultural fields and other areas before the practice was prohibited in Vermont.
- Bennington Landfill, including sludge disposed there from the Bennington WWTP.
- Other industrial facilities that used PFAS or materials containing PFAS in their operations.

This report also presents the results of a preliminary evaluation of some potential sources of PFAS other than the former Chemfab facilities within the study area. These sources include commercial, industrial, and other properties throughout the area.

Measured PFOA levels in soil, surface water, groundwater, and landfill leachate are as follows:

- Concentrations in soil samples range from no detection to 45 parts per billion (ppb). All samples were below the Department of Health's soil screening value (SSV) for PFOA of 300 ppb.
- Concentrations in surface water samples range from no detection to 79 parts per trillion (ppt).
- Concentrations in groundwater samples range from no detection to 5,600 ppt at the former Chemfab facility on Water Street. The highest concentration in a private well sample is 4,600 ppt.
- Concentrations in leachate at Bennington Landfill range up to 5,300 ppt.

For purposes of this report, the following potential sources and pathways for PFOA were evaluated:

- Emissions of PFOA through stacks at the two former Chemfab facilities. PFOA emissions were estimated based on annual dispersions usage and a conservative (a cautious approach resulting in higher than expected conditions) estimate of PFOA content in the dispersions of 2,000 parts per million (ppm). Modeled annual emissions for the Northside Drive facility were 47 pounds per year on average (1969 – 1978), and modeled annual emissions for the Water Street facility were 145 pounds per year on average (1978 – 2002). Emissions were assumed to be continuous throughout the operating period of each facility.
- Deposition of PFOA from air to the ground surface. Hourly variations in wind speed and direction were simulated along with the variation in ground surface elevation over the air model domain.
- Dissolution in water infiltrating the ground surface and transport in water through the unsaturated zone to the saturated zone at the water table. Spatial variations in infiltration rates were simulated.
- Transport in the unconsolidated material and bedrock throughout the region potentially affected by air deposition from the former Chemfab facilities.

Findings of this evaluation include the following:

- Airborne emissions of PFOA from the former Chemfab facilities may have contributed to PFOA in groundwater in portions of the Bennington area. They cannot be the source of PFOA in other areas where PFOA has been detected in wells at concentrations above Vermont's drinking water health advisory level of 20 ppt, including areas to the south and southwest of Bennington Landfill.
- Transport through the unsaturated zone likely produced a lag between the time PFOA was deposited at the ground surface and the time PFOA reached the water table.
- Simulated concentrations of PFOA in groundwater range from 0 ppt up to approximately 1,000 ppt in close proximity to the former Water Street Chemfab facility.
- The modeled processes and values used as model inputs explain certain patterns of PFOA concentrations, as well as illuminate areas where the air deposition and groundwater transport processes do not explain the presence and distribution of PFOA in wells and where other sources, such as landfills or other activities, are a more likely source for PFAS detections.
- The modeling indicates that the presence of PFOA in groundwater in the vicinity of Bennington Landfill is not the result of air emissions from the Chemfab plants. These results suggest that

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sources at the landfill are the likely source of PFOA and PFOS detected in wells south and southwest of Bennington Landfill.

- There are numerous other potential sources of PFAS in the study area that were not included as PFAS sources for purposes of this report but which could explain PFAS groundwater impacts. These potential sources include a wide variety of commercial and industrial sites (e.g., car washes; landfills, granite, stone and tile finishing; storage and/or use of fire-fighting foam; wastewater treatment plants; wood floor finishing; automotive repair; junkyards; incinerators; carpet and upholstery cleaning; and painting/coating application); and sites that have used biosolids (e.g., agricultural fields; athletic fields; turf farms; landscaped areas; greenhouses; nurseries; and landscape and topsoil supply facilities).

Preliminary modeling results presented in previous versions of this draft report (e.g., Barr, 2017) have been used to delineate Corrective Action Area 1 within the study area. Corrective Action Area 1 (CAA1) is defined as the area in which the conceptual modeling results indicate that air emissions from the former Chemfab facilities are a potential source of PFAS concentrations in groundwater exceeding the associated drinking water standard (i.e., PFOA+PFOS = 20 ppt).

## 2.0 Introduction

### 2.1 Situation Statement and Purpose

An area with exceedances of regulatory standards for perfluorooctanoic acid (PFOA) in soil and groundwater has been identified near two former Chemfab facilities in North Bennington and Bennington, Vermont. This report describes some of the available information and conceptual modeling analyses performed to evaluate the potential for PFOA release mechanisms and transport pathways from the former Chemfab facilities to provide a mechanistic evaluation of the distribution of measured PFOA and other perfluorinated (PF) compounds in certain of the wells in the area.

For purposes of this report, the simulations are not intended to account for the concentrations measured in *individual* wells. Data that are typically available for simulating solute transport, such as water levels in most affected wells, stream gaging data, aquifer tests, etc., are not currently available for the Bennington area. However, for purposes of this report, the available data and information that is relevant to the processes involved in PFOA fate and transport were utilized and there is sufficient information and understanding of mechanisms to develop higher-level conceptual models that are useful in evaluating the data and to inform further investigation.

### 2.2 Characteristics of PFOA, PFOS, and Associated Perfluoro Compounds

PFOA (perfluorooctanoic acid) is one of a class of fluorinated hydrocarbon compounds associated with fluoropolymers – most commonly recognized as Teflon®. Like PFOA, perfluorooctanesulfonic acid (PFOS) is a “C8” perfluoro (PF) compound, possessing eight carbon atoms, each attached to two fluorine atoms. The carbon-fluorine bond is one of the strongest molecular bonds and results in characteristics that are useful in the production of chemical, electrical, and heat-resistant materials. PFOA was manufactured by 3M and DuPont in the United States until their production was phased out in 2002-2006. PFOS was manufactured by 3M using an electrochemical fluorination process until 2002. Since that time, shorter chain perfluorocarbons have been used.

PFOA and PFOS were used in the manufacture of many commercial materials for industrial and residential use, including stain-resistant carpeting/furniture/upholstery, non-stick cookware, food package coatings, aqueous fire-fighting foams (AFFFs), moisture-resistant breathable fabrics, concrete and rock sealants, electrical capacitors, dyes, paints and coatings, batteries, photographic films, printing inks, herbicide and pesticide formulations, car wash surfactants, and as a vapor suppressor in metal plating processes (Kissa, 2001; 3M, 1999; Knepper and Langue, 2012). They are a ubiquitous presence in most households and consequently are found in municipal waste streams and in most landfills (Busch et al., 2009; MPCA, 2009). They are also found in biosolids (in part from municipal sewer sludge, e.g. Sepulvado et al., 2011) used as amendments to soil in agricultural and landscaping applications, including gravel pit reclamation (NEBRA, 2014).

PFOA and PFOS are highly soluble in water and are resistant to degradation. They adsorb poorly to material with low organic material content, and where they bond, it is to surfaces of organic material. In saturated, unconsolidated deposits and fractured rock media, they are mobile and migrate as an unattenuated solute in flowing groundwater. In general, they are considered "conservative" solutes because the compounds' conservation of mass is confined to the dissolved phase and the compound is not substantially converted to other forms. Very small amounts of PFOA and PFOS can result in detections in groundwater in the parts per trillion (ppt) and parts per billion (ppb) range. They are typically detected downgradient of landfills as a result of leaching of PFAS from consumer and commercial materials disposed of in the landfills, such as treated carpeting, fast-food wrapping, floor wax, composted materials, waste-water treatment plant sludge, dyes, textiles, tape, batteries, and furniture (Schaider et al., 2017). For example, Minnesota has required sampling and analyzing for PFOA and other PF compounds at open and closed landfills since 2007 (MPCA, 2009); PFOA and PFOS have been found in downgradient monitoring wells at most landfills. PFOA attributed to leachate from landfills has been found in groundwater in the Netherlands (Eschauzier et al., 2013), Germany (Busch et al., 2009; Stahl et al., 2012), Canada (Li et al., 2012), Finland (Perkola and Sainio, 2013), and undisclosed locations throughout Europe (Eggen et al., 2010). Typically, PFOA (and PFOS) will be detected in wells where other landfill-related constituents (e.g., volatile organic compounds, chlorinated compounds, chloride, and metals) are not detected, indicating both their high level of mobility and the extremely low detection levels that are necessitated by sampling protocols. If specialized analytical procedures are used to detect very low levels of conventional landfill constituents (e.g. volatile organic compounds), they too may be detected in wells that have low concentrations of PFOA.

PFAS in wastewater treatment effluent/sludge and septage represents the contributions from a wide range of commercial, domestic, and industrial sources. PFAS have been found to be widespread in wastewater discharge and have been shown to concentrate in the high organic content environment of wastewater treatment plants (Schultz et al., 2006). Therefore, PFAS are commonly found in wastewater treatment sludges (Zareitalabad et al., 2013).

Perfluoroalkyl carboxylates, the class of PFAS containing PFOA, are stable in water, sediment, and soil under natural conditions (Kissa, 2001). As a result, longer-chained perfluoroalkyl carboxylates are not known to degrade into PFOA in these media in the environment. PFOA is itself a degradation product of perfluoro alcohols. The perfluoro alcohols that degrade to PFOA are volatile and may be emitted into the atmosphere as part of various chemical manufacturing processes. Consequently, PFOA is found worldwide at low levels and there is a background level in the aqueous environment virtually everywhere in the world.

In some manufacturing processes, PFOA is used and may be released to the environment through air emissions. Production of PFOA-coated textiles is an example of such a process. Stack testing results for the Saint-Gobain facility in Merrimack, New Hampshire show that the PFOA is primarily emitted as a fine particulate (less than 1- $\mu$ m diameter; Barton et al., 2006) that flows out of stacks and migrates downwind. PFOA is deposited on the ground in proximity to the emission source.

PFAS with carbon-fluorine chains longer than PFOA and PFOS (C8) were detected in soils in close proximity to the former Water Street Chemfab facility. Perfluorotetradecanoic acid (PFTeDA or C14) was detected at 11 of 43 sampling locations near the facility, ranging from 1.4 ppb to 49 ppb in the 0 to 6 inch sample interval (C.T. Male Associates, 2016). Concentrations decrease to single digits below the 0 to 6 inch interval. At the former Chemfab facility, PFTeDA was detected in soil at six sampling locations and at concentrations as high as 93 ppb, but was not found in groundwater samples in this area (Table 1).

Several studies have quantified the structure-activity relationships between the length of linear PFAS and transport potential through soil columns (Higgins and Luthy, 2006; Zhang et al., 2013; Gellrich et al., 2012; Hirata-Koizumi et al., 2015). PFTeDA and other longer-chained fluorocarbons also have a much higher partition coefficient with respect to organic carbon, resulting in very low mobility in the soil column which prevents them from migrating to groundwater (which likely explains why they were not detected in deeper soils or groundwater where they were detected in shallow soils). Thus, compared to PFOA, PFTeDA and other long-chain linear PFAS found in organic soils are less likely to affect groundwater because of their inability to percolate through soil columns. These compounds likely represent trace constituents in the dispersions used at the facility and were deposited on the roof, where they washed to nearby soil as roof runoff (the highest concentration of PFTeDA was found next to a roof drain). Because they are very immobile, they have remained in the upper soil zone where they were deposited.

PFAS found in organic soils present a small risk to groundwater supplies, because of their inability to percolate through soil columns.

### 2.3 PFOA Study Area

The area near the former Chemfab facility in North Bennington has been identified by the Vermont Department of Environmental Conservation (VTDEC) as an area of interest in the investigation of PFOA concentrations exceeding regulatory standards in groundwater. The study area, which is shown on Figure 1, is centered approximately on the former Chemfab facility on Water Street in North Bennington and includes all of the Village of North Bennington and portions of the Town of Bennington.

The investigation, to date, has found PFOA at detectable concentrations in certain soil borings and groundwater monitoring wells at the former facility and in soil and groundwater at certain locations in North Bennington and the Town of Bennington, as shown on Figure 1. PFOA has not been detected in the public water supply systems for North Bennington and Bennington, which are sourced beyond the study area. Details on water supply sources and locations are provided in Section 3.8. PFOA concentrations in soil samples are below the Vermont Department of Health's soil screening value (SSV) of 300 ng/g (ppb). PFOA concentrations are generally higher in groundwater samples from monitoring wells and private wells at and near the former Chemfab facility on Water Street in North Bennington or in the vicinity of the former Bennington Landfill. In some wells, PFOA levels exceed the Vermont Department of Health's drinking water health advisory (DWHA) of 20 ng/L (ppt). Details on PFOA concentrations are provided in Section 4.

## 2.4 Potential PFOA Sources in the Study Area

A potential source of PFOA is a facility or activity with a known or likely release of PFOA to the environment at sufficient volume to result in detectable concentrations of PFOA in soil and/or groundwater. Examples of these sources range from industries using PFOA as part of their manufacturing process to fire stations, wastewater treatment plants, and airports at which aqueous fire-fighting foams would be dispensed.

Multiple potential sources of PFOA are within the study area (Figure 1). These include the former Chemfab facilities in North Bennington (Water Street) and Bennington (Northside Drive); landfills and solid-waste disposal areas where PFOA has been confirmed in groundwater monitoring wells (e.g., Bennington Landfill, Kocher Drive Landfill); former dump sites where disposing of miscellaneous waste (including waste potentially containing PFAS) may have occurred (e.g., on and near Bennington College property); and areas in which sewage sludge from the Bennington Wastewater Treatment Plant (WWTP) was disposed, such as Bennington Landfill. Additionally, there are a number of industries in the area that likely used PFOA or other PFAS in their processes.

### 2.4.1 Former Chemfab Facilities

Two former Chemfab facilities are located within the study area (Figure 1). The former Chemfab facility on Northside Drive in Bennington operated from 1969 to mid-1978 when operations were transferred to the Water Street facility in North Bennington. The Water Street facility operated from mid-1978 through February 2002 when the plant was closed. These facilities applied PTFE coatings to fiberglass fabrics. During drying and curing of the PTFE, PFOA driven off of the fabric may have been emitted from the facilities' stacks. A discussion of emissions, air transport, and deposition of PFOA is included in Section 5.2.1. Additional potential sources at the facilities include roof drip lines and locations where direct deposition of materials containing PF compounds may have taken place.

### 2.4.2 Landfills and Disposal Areas

Bennington Landfill, shown on Figure 1, is a closed landfill that received municipal sanitary waste from the Town of Bennington from 1969 until 1987 (USEPA, 2014). The landfill has also been referred to as the Former Bennington Landfill and the Houghton Lane Landfill. Historical records indicate that polychlorinated biphenyls (PCBs), volatile organic compounds (VOCs), and lead were disposed of in the landfill. Several industries in the Bennington area dumped liquid wastes into an unlined lagoon at the landfill from 1969 to 1975. An underdrain was installed in 1976 to dewater the unlined lagoon (McLaren/Hart, 1997). This underdrain was extended to the north in 1979 to 1980 as waste disposal occurred in that area. A treatment system was eventually installed to treat leachate collected by the underdrain. The treated leachate was infiltrated onsite. The landfill was capped and graded and other measures were taken to reduce production of leachate. The leachate treatment system was decommissioned in 2008. Since 2008, untreated leachate has been collected in a storage vault, pumped out of the vault when leachate levels are high, and treated and disposed offsite (VT ANR, 2016).

Leachate from Bennington Landfill has been shown to have contaminated surface water and groundwater with PCBs and VOCs. The facility was listed as a Superfund site in 1989 for this reason (USEPA, 2004).

Constituents monitored in the landfill leachate and monitoring wells include PCBs and VOCs. Monitoring for PFOA and other PFAS was not undertaken until an investigation initiated by VTDEC in early 2016. Detected concentrations of PFOA at the landfill are discussed in Section 4.

Other potential sources include various disposal areas and landfills in the area. VTDEC has identified the Kocher Drive Landfill (a.k.a. Kocher Drive Dump, shown on Figure 1) as having monitoring wells with PFOA. There are also potential former disposal areas on and near Bennington College that could contain sources of PFAS, which were identified from historical air photos and accounts of nearby residents. Landfill materials typically contain consumer and commercial goods that have varying amounts of perfluorochemicals, such as treated carpeting, fast-food wrapping, floor wax, composted materials, wastewater treatment plant sludge, dyes, textiles, tape, batteries, and furniture (Schaider et al., 2017)

#### **2.4.3 Wastewater Treatment Plant Sludge Disposal Areas**

In other areas with documented PFOA detection, PFOA has been detected in the sewage sludge produced during wastewater treatment (USEPA, 2014). Stahl et al. (2012) report that wastewater sludge that was spread on agricultural lands in Germany contained PFAS that leached into groundwater. Sludge from the Bennington WWTP has reportedly been composted and land-applied, as discussed in Section 5.1.3. Locations of this land application are not currently known. It is also possible that livestock drank water from groundwater or surface water supplies that contained PFOA and subsequently deposited PFOA onto land surfaces via manure and urine. Similarly, human consumption of PFOA-containing water can result in redistribution of PFOA through discharge from private septic systems.

Sludge from Bennington's WWTP was reportedly stockpiled for short periods of time in the western portion of Bennington Landfill (TRC, 1995; TRC, 1997; McLaren/Hart, 1997). While analytical data confirming the presence of PFOA in sludge stockpiled at the Bennington Landfill is not available, research indicates PFAS are commonly found in wastewater treatment sludges (Zareitalabad et al., 2013). In addition, groundwater monitoring wells in the vicinity of the sludge disposal areas are believed to be contaminated with metals from the sludge (McLaren/Hart, 1997) confirming impact to the subsurface from these stockpiling activities. Plans at that time called for the stockpiled sludge to be utilized as a component of the landfill capping material.

Anecdotal information supplied by VTDEC indicates that sludge from the Bennington WWTP may have been supplied to local farmers for land application. Follow-up evaluation is needed to determine whether sludge from the WWTP was land applied in the Bennington area and, if so, where it was applied.

#### **2.4.4 Industrial Users**

There are several industries in the area that likely used PFAS in their processes. Investigation into potential sources of PFAS is ongoing.

An inventory of wastes disposed of at Bennington Landfill provides some insight into their waste streams and the potential for their use and/or disposal of PFAS. These industrial entities include the following:

1. **Ben-mount Corporation (Textron, Inc.).** The Ben-mount Corporation used paper print, ink cleaning solvents and reported disposal of 772,200 gals of waste to the Bennington landfill. Paper coatings and some inks and solvents contain PFAS.
2. **B. Co. (formerly known as (f/k/a) Bijur Lubricating Co.)** Lubricating oil, mist coolant, and grinding sludge from this facility were reported disposed of at the Bennington Landfill [290,000 gals]. PFAS are used in some hydraulic fluids and mist suppressants.
3. **Catamount Dyers.** Catamount Dyers disposed of 168,000 gallons of waste oils and dyes at the Bennington Landfill. Some dyes are known to contain PFAS that may be related to textile treatment.
4. **Courtalds Structural Composites, Inc.** Courtalds Structural Composites disposed of an unspecified quantity of oils, metal chips, paint, resins, and thinners. Disposal from Courtalds also included unspecified quantities of "Teflon® Film" (Vickers, 1990). Some oils, paints, resins, and thinners are known to contain PFAS.
5. **Eveready Battery Company, Inc.** 336,000 gallons of small chemical batteries and lighting products were disposed of at the Bennington Landfill. Lithium-ion batteries contain PFAS.
6. **Jard Company, Inc.** Jard Company manufactured capacitors using PCBs and disposed of 760,000 gallons of waste in the Bennington Landfill. Electrical components use PFAS in various parts and production.
7. **Johnson Controls Battery Group, Inc.** Lead acid batteries were manufactured by Johnson Controls and 575,000 gallons of waste were disposed of at the Bennington Landfill. Plating operations use PFAS as mist suppressants.
8. **Arken Industries, Inc.** disposed 30,000 gallons of metal chips, cutting oils, and solvents at Bennington Landfill and may have used PFAS.
9. **MascoTech Controls, Inc. (f/k/a Schelzer Corporation)** were manufacturers of vacuum brake assemblies and contributed 17,000 gallons of waste to the landfill. PFAS are used in some hydraulic fluids.
10. **Sibley Manufacturing Co. Inc. / CLR Corporation** is a machine tool company with waste products of metals chips, cutting oils, and solvents. The company contributed 28,000 gallons of waste to the landfill and may have used PFAS.
11. **Triangle Wire & Cable Inc.** was a cord set assembler that disposed of 17,000 gallons of waste oil at the Bennington Landfill. PFAS are used in some wire and cable coatings.

For comparison, Chemfab is reported to have contributed 14,000 gallons of waste (some of which reportedly contained PTFE-containing dispersions) to Bennington Landfill.

Nineteen other entities each contributed between 150 and 14,000 gallons of waste to the landfill, in areas of steel and iron fabrication, stoneware, aerospace, tools manufacture, dairy products/pasteurization, photoplating, fuels, machine shop, transportation of liquid and dry bulk, plastic bags manufacture, plastics, hospital, college, sprocket manufacture, and storage of vehicles and equipment. The known associated wastes include print/ink, Teflon® film, asbestos, and packing tissue.

For this evaluation, the above industries and potential disposal areas (operating and former) were not simulated as potential sources of detection of PFOA.

## 2.5 Conceptual Modeling Analyses

For purposes of this report, numerical models were developed to simulate PFOA transport and migration from points of release to the groundwater flow system and private wells. Given the limited amount of hydrogeologic data and some uncertainties in sources of PFOA, the modeling approach constitutes a conservative, regional evaluation. The simulations allow for a quantitative and physically-based assessment of the mechanisms that could potentially result in detectable PFOA concentrations in soils and groundwater in areas at distance from the release points.

### 2.5.1 Overview of Approach

For purposes of this report, the complete PFOA transport pathway was simulated to assess the possibility of transport from source to potential receptors. As indicated above, these transport paths are through air, unsaturated soil and other unconsolidated materials, and saturated aquifer materials.

No single model in standard practice exists that can account for transport through these different media. Therefore, transport was simulated by a series of distinct models, with the output of one or more models serving as input for the model simulating the next segment of the transport pathway. Each of these models is a widely accepted tool for the transport pathway for which it was applied.

### 2.5.2 Assumptions and Limitations

The available data and standard model input values were incorporated in the modeling to the extent that such information was available in order to meet the purpose of this report. Modeling decisions that were made and their basis are discussed in the modeling sections below and the supporting appendices detailing model setup.

As with all deterministic models of natural processes, these model results are non-unique: the result is one of many possible outcomes. Given the uncertainty in parameter values and the available data quality, standard calibration procedures could not be used to the extent that would be employed if more information and data were available. Additional data collection activities planned as part of the CSM Site Investigation will provide data to reduce uncertainty and to test hypotheses.

## 3.0 Hydrogeologic Setting

### 3.1 Summary

North Bennington is located in the Northeastern Appalachians groundwater region (Randall et al., 1988), and the following summary is condensed from the indicated resource. This region is characterized by rolling topography that primarily reflects the weathered bedrock surface with glacial and fluvial landforms mantling the bedrock. The bedrock consists of folded and faulted metamorphosed sedimentary rocks with low primary porosity. Water is conducted in the bedrock through secondary porosity. In some rock types, such as those in the North Bennington area, the secondary porosity is solution enlarged. The frequency and permeability of secondary porosity typically decreases with depth. Most bedrock wells in the region are up to 200 to 400 feet deep. While geology is a controlling factor of groundwater processes, geologic and hydrogeologic conditions are not synonymous and hydrogeologic parameters and mechanisms are often not correlated with geologic units (particularly bedrock geologic units). Geology is an important aspect of groundwater flow but it is the "container" in which groundwater flow processes take place and not the driver for flow direction. Sources and sinks ("divergences") for groundwater generally dictate groundwater flow direction. Recharge from infiltrating precipitation is nearly always the most important source and discharge to rivers is nearly always the most important groundwater sink. Characterizing recharge and discharge is the key to understanding groundwater flow directions.

Glacial erosion and deposition produced changes in drainage and topography and deposited a nearly continuous layer of unconsolidated till over the bedrock. Stratified drift units, chiefly sand and gravel, follow the larger valleys such as those of the Walloomsac River.

Recharge to the bedrock is controlled by the permeability and thickness of the overlying glacial deposits and overburden. Runoff in upland areas is focused to seasonal streams that typically lose discharge in areas in which they flow over stratified drift at the margins of the larger valleys. Recharge to bedrock wells that are pumped continuously may occur from adjacent stratified drift aquifers. Discharge is primarily from the bedrock to wells and to the stratified drift in the large valleys. Inter-basin flow systems with significant discharge have not been discovered in the bedrock.

In addition to discharge to the stratified drift filling the larger valleys from minor upland streams, recharge to these units is from direct infiltration of precipitation, and discharge from bedrock. Recharge from the larger rivers may take place in the case of localized pumping from the stratified drift or higher-than-normal river stage. Discharge from the stratified drift is via pumping wells, evapotranspiration, and to the larger rivers when their stage is at or below typical levels.

### 3.2 Surficial Materials

Surficial materials in the study area were deposited in the bedrock valleys by glacial processes and modern alluvial processes. The materials are characterized by lithological heterogeneity that reflects the various depositional settings. Lithology ranges from fine-grained tills to coarse-grained outwash and alluvium. The distribution of unconsolidated materials is shown on Figures 2A and 2B. (Note that the interpretation on Figure 2B from DeSimone (2017) was recently completed and was not determined to be

sufficiently different from previous interpretations to warrant modification of the modeling work described in this report.) In general, fine-grained tills blanket much of the area and coarse-grained surficial materials are found adjacent to surface water features (Figure 2).

The thickness of surficial materials is variable, ranging from absent at bedrock outcrops to more than 300 feet (Figure 2). In general, surficial material thickness in the North Bennington area is greatest in bedrock valleys north of the Walloomsac River and decreases approaching the Green Mountains to the east. At the Water Street Site in North Bennington, soil boring data indicated the possible presence of a buried bedrock valley with depth to bedrock greater than 60 feet.

For the purpose of this report, the surficial materials are considered a single hydrogeologic unit. A second unconsolidated aquifer system has been identified in sand and gravel units confined beneath a thick layer of till in the area of Bennington and Shaftsbury, Vermont (Bennington-Shaftsbury area; Jerris and DeSimone, 1992). These deposits are inferred to be continuous in the deepest portions of the bedrock paleo-valleys, but may be locally discontinuous. Well yields from these deposits range from 5 to 100 gallons per minute. Well-log data suggest that these deposits are recharged by the bedrock fracture systems and the recharge zones for the confined sand and gravel aquifer are believed to be the same as for the bedrock aquifer system (Jerris and DeSimone, 1992). Due to their inferred hydraulic connection with the underlying bedrock aquifer, these units were lumped with the bedrock as a single hydrostratigraphic unit. The zonation for hydraulic conductivity in layer 2 of the model currently does not include zones representing these confined sand and gravel units.

The confined sand and gravel units likely have different hydraulic characteristics than the bedrock aquifer, however, no data are available to quantify these differences. The confined sand and gravel units have been identified in a small area of the model domain, and are not well defined. The confining layer separates these confined sand and gravel units from the discharge zone, thus likely prevents these units from affecting the transport characteristics of the bedrock. For these reasons, the influence of these units on model outcomes would likely be negligible.

According to Jerris and DeSimone (1992), approximately 3.5 percent of the wells in the Bennington-Shaftsbury area are completed in the unconfined sand and gravel deposits. This interpretation is based on the available well logs, which would cover only a fraction of the wells completed at the time and none of the wells completed since. More recent evaluations of aquifer completion made by VTDEC as part of private well sampling put the percentage of wells completed in shallow, non-bedrock units (either confined or unconfined sands and gravels) at closer to 20 percent (based on data collected as of November 11, 2016). Yields from wells completed in the unconfined sand and gravel deposits range from 5 to 60 gallons per minute. Recharge enters this aquifer system in the form of direct precipitation. The unconfined nature of this aquifer system makes it particularly susceptible to water-quality impacts (Jerris and DeSimone, 1992).

### 3.3 Bedrock

Bedrock underlying the PFOA study area consists of sedimentary and metamorphosed sedimentary rocks, including limestone, phyllite, gneiss, and marble (Figure 3). The metamorphic rocks have been faulted and

folded to varying degrees by structural events primarily during the Taconic and Acadian Orogenies (Spencer, 1977).

Groundwater flow in the bedrock is primarily through secondary porosity features (i.e., those that were created after the rock was formed). The secondary porosity features in bedrock within the study area include fractures formed by weathering and structural forces and voids formed by dissolution of carbonate minerals.

Jerris and DeSimone (1992) reported that approximately 73 percent of the wells in the vicinity of Bennington-Shaftsbury area are completed in bedrock. The bedrock aquifer is most productive in carbonate rock and less productive in quartzite, granite/gneiss, and phyllite/schist lithologies. Bedrock wells providing adequate yields in this area range in depth from 100 to 300 feet, but they may be deeper in carbonate units. Recharge was inferred by these authors to enter the bedrock at elevations above 1,200 feet where till deposits are thin and scattered and bedrock fracture systems are directly exposed to precipitation and runoff.

Groundwater flow in fractured bedrock is in conduits and not primarily in the pore spaces between granular materials (as is the case with unconsolidated deposits). Conduit flow does not adhere to Darcy's Law – the governing principle in groundwater flow mechanics. At a very local scale, groundwater flow direction in fractured rock can be counter to regional flow directions. As the scale of observation increases, groundwater flow in fractured rock aligns with regional flow because many more of the fractures and conduits become interconnected and groundwater flow in the fractured rock behaves more like a porous media, adhering to Darcy's Law. Fractured rocks are treated as "equivalent porous media" (EPM) at scales larger than the Representative Elemental Volume (REV), with the REV being the smallest volume in which fracture interconnection is sufficient for Darcy's Law to be descriptive of groundwater flow. In practice, the EPM treatment of fractured rock for regional evaluations is almost always proven to be sufficient. Characterizing and simulating conduit flow or employing such modeling approaches as discrete fracture networks are extremely data intensive (Neville and Bedekar, 2016). Horizontal anisotropy may be simulated if fractures or other factors such as dipping strata are known to create a preferred direction of flow (Anderson, Woessner, and Hunt, 2015). Such preferred directions of flow are not known to occur at scales comparable to that of the model domain.

### 3.4 Hydrostratigraphy

A hydrostratigraphic unit is one or more geologic units that are distinct from the surrounding materials and that share similar hydraulic properties. Hydrostratigraphic units are typically classified as aquifers or aquitards, depending on the amount of groundwater they yield. Assigning geologic units to hydrostratigraphic units provides a geologic interpretation that represents important hydrogeologic characteristics, such as the resistance to groundwater flow. Multiple geologic units may be combined into a single hydrostratigraphic unit, and a single geologic unit may be divided into one or more hydrostratigraphic units.

For purposes of this report, the unconsolidated surficial materials and the fractured bedrock are considered to be distinct hydrostratigraphic units. Jerris and DeSimone (1992) subdivided the surficial

material into a shallow, unconfined sand and gravel aquifer and a deep, confined sand and gravel aquifer. Such a subdivision was not adopted for this evaluation for reasons described in Section 3.2. The confined sand and gravel units likely have different hydraulic characteristics than the bedrock aquifer, however, no data are available to quantify these differences. The confined sand and gravel units have been identified in a small area of the model domain, and are not well defined. The confining layer separates these confined sand and gravel units from the discharge zone, thus likely prevents these units from affecting the transport characteristics of the bedrock. For these reasons, the influence of these units on model outcomes would likely be negligible.

### 3.5 Groundwater-Surface Water Interactions

In general, the groundwater flow system in the study area is topography-controlled, meaning the water table can be roughly conceptualized as a reflection of the ground surface (Randall et al., 1988). Topography-controlled water tables develop in areas with relatively low-permeability aquifers subjected to relatively high rates of recharge; in addition, the distance between surface water features is an important factor (Haitjema and Mitchell-Bruker, 2005). In topography-controlled aquifers, water table lows form along brooks and rivers where groundwater discharge takes place and water table highs form between brooks and rivers due to the mounding effects of recharge.

The streams and rivers within the study area generally act as zones of groundwater discharge (sinks). The former Chemfab facility on Water Street is located in North Bennington along the western bank of Paran Creek. Paran Creek discharges to the Walloomsac River, which is the primary drainage feature in the study area.

For purposes of this report, the modeling represents long-term average conditions. Temporary losing reaches may result from river stage fluctuations, but would be expected to affect only the area in close proximity to the river. The process of hydraulic gradient reversal (and temporary transition from gaining to losing) results in temporary bank storage, which has been shown to affect groundwater chemistry in only the immediate area of the river (e.g., within 30 meters horizontally and 4 meters vertically; Squillace, 1999).

Given the controls of topography on the groundwater flow system in the study area, the Walloomsac River and other major streams and rivers are ultimately discharge points for the groundwater flow system, regardless of periodic reversals in flow between the river and aquifer. Therefore, the effects of bank storage would not be expected to affect the model outcome.

The relatively high contribution of groundwater to the flow in rivers and streams in the study area is supported by an assessment of baseflow. Baseflow is the component of streamflow that is sustained by groundwater discharge to the stream, and the baseflow index for a streamflow measurement location is the proportion of total streamflow that is baseflow. As part of this evaluation, a baseflow index (BFI) of 0.71 was estimated for the Walloomsac River at North Bennington (USGS Gage 01334000) by hydrograph separation using Purdue University's Web-Based Hydrograph Analysis Tool (Lim et al., 2005). This value indicates that approximately 70 percent of the total streamflow measured at the gaging station originates as baseflow. Groundwater in the area flows to the Walloomsac River, where it is transported out of the area and becomes part of the regional river system. Therefore, PFOA in the groundwater will also flow into

the Walloomsac River, where it will become diluted below detectable levels and transported as stream flow out of the area.

### 3.6 Recharge and Infiltration

The primary source of groundwater recharge in the surficial and bedrock aquifers is infiltrating precipitation. Only a portion of the direct precipitation infiltrates below the land surface and only a portion of the infiltrated precipitation recharges the groundwater system. Runoff, evaporation, and plant water uptake (transpiration) can account for a significant portion of the direct precipitation and act to send the water downslope or back to the atmosphere.

The amount of direct precipitation that infiltrates the land surface and moves below the root zone is the maximum amount of water available to recharge the groundwater system. This amount is dependent upon the rate and duration of precipitation, the soil type, land cover, land use, evapotranspiration, and topography.

Strictly speaking, infiltration is similar to but differs from groundwater recharge. The most important distinction between infiltration and recharge is the time lag between infiltration of water past the root zone and recharge at the water table. In addition, small-scale processes such as local flow systems and rejection of infiltration due to saturated soils at the ground surface often result in differences between recharge and infiltration. Despite these differences, calculated infiltration rates and groundwater recharge are considered to be approximately equal and are treated as such in this evaluation.

Jerris and DeSimone (1992) provided a range of aquifer recharge values of 20 to 35 inches per year, depending on elevation. These estimates were based on differences between precipitation records and stream gaging records for the Green River in Williamstown, Massachusetts. Average annual recharge of 21 inches per year was inferred from streamflow records in basins throughout Vermont and New Hampshire for the period of record from 1961 to 1990 (Flynn and Tasker, 2004).

### 3.7 Groundwater Levels and Flow Directions

In general, the groundwater system is topography-controlled (Randall et al., 1988), meaning the water-table elevation is a reflection (lower-relief replica) of the ground surface. Inferred flow directions in the bedrock and unconsolidated materials are shown on Figure 4. In topography-controlled settings, the boundary of the "groundwatershed" can be inferred to generally coincide with the hydrologic watershed boundary. The rivers and larger streams are discharge zones. In general, groundwater does not flow across rivers and streams unless there is a substantial pumping center or other discharge source near the stream.

Groundwater flow is from the areas of recharge to the areas of discharge; in this case, from the upland areas to the rivers and streams. Depth to groundwater varies across the study area, with a median value of 30 feet from ground surface for the private wells within the domain of the air model described in Section 6.1.1.

Hydrogeologic studies at Bennington Landfill have shown locally perched conditions in the shallow groundwater system beneath the landfill (see Section 5.2.4). The result is that the groundwater flow direction in the shallow groundwater system impacted by the landfill is different from that in the bedrock flow system (Figure 4). Groundwater flows primarily east and then south in the overburden in the vicinity of the Bennington Landfill. Groundwater in the underlying bedrock flows primarily south-southwest to the Walloomsac River. Along the flow path in the overburden, there is also the potential for downward leakage into the bedrock and subsequent southwesterly flow.

Groundwater flowing through the bedrock fracture network entered the system either as vertical leakage through the overlying surficial materials or as infiltrating precipitation in bedrock exposures (i.e., outcrops). Flow through the bedrock is through secondary porosity features (i.e., fractures, joints, bedding planes, and karst). The degree of fracturing is not known, though is expected to decrease with depth. Karstification has not been observed from the available data, but has been previously interpreted to be a controlling feature of the groundwater system discharging at Morgan Spring in downtown Bennington (Town of Bennington, 2016).

### 3.8 Groundwater Withdrawal

Groundwater withdrawal in the study area is primarily from private wells. Individually, private wells do not have high groundwater withdrawal rates but in aggregate and in close proximity to one another, may affect groundwater flow directions locally. No high-capacity industrial or commercial wells were identified from available information on water use and well construction. Municipal water supplies for North Bennington and Bennington have a significant surface water component. As discussed in Section 4.2.3, PFOA has not been detected in these municipal water supplies.

In general, groundwater flow in the study area is a reflection of the land surface topography, which is the most dominant control on groundwater flow directions. The short duration and relatively low rates of pumping of private wells is not anticipated to affect overall groundwater flow directions in the area, due to the localized cones of depression that would result. Also, the density and close spacing of private wells would serve to cancel out the effects on flow paths, due to a perpetual push-pull from drawdown between nearby wells.

The water sources for Bennington are Morgan Spring and Bolles Brook, which are spring and stream source types, respectively (Town of Bennington, 2016). Morgan Spring is a high-yielding spring water supply located in downtown Bennington (Figure 1). The intake on Bolles Brook is in the Town of Woodford, approximately 6.5 miles east of the former Chemfab facility in North Bennington. Morgan Spring is classified as a groundwater source according to EPA source type, but would not be considered a groundwater withdrawal because it is a natural groundwater discharge feature.

North Bennington's water supply is sourced from surface water and groundwater (VTDEC, 2016). Groundwater is pumped from supply wells near the surface water intake on Basin Brook that is located in Shaftsbury, approximately five miles northeast of the former Chemfab facility in North Bennington and approximately two miles beyond the edge of the PFOA study area delimited by VTDEC (Figure 1). The wells are completed in alluvial material associated with Basin Brook where it flows from the Green

Mountain National Forest. There is no distinction between groundwater and surface water sources in the water use data reported to VTDEC's Drinking Water and Groundwater Protection Division.

## 4.0 Current PFOA Concentrations

Concentrations of PFOA in soil, groundwater, and surface water have been measured in 2016 as part of investigations conducted by Saint-Gobain, VTDEC, and the U.S. Environmental Protection Agency (EPA). Soil sampling results show PFOA concentrations below the applicable Vermont soil screening value of 300 ppb. Detections of PFOA at monitoring wells at the former Chemfab facility on Water Street, in certain private wells in North Bennington and the Town of Bennington and at Bennington Landfill exceed the Vermont Department of Health's DWHA of 20 ppt.

### 4.1 Soil and Wastewater Treatment Plant Sludge

Soil samples collected from multiple depths at locations surrounding the former Chemfab facility on Water Street in North Bennington show PFOA concentrations ranging from no detection to 45 ppb (Figure 5; C.T. Male Associates, 2016). The majority of samples contain PFOA at concentrations below 10 ppb. Levels on the former Chemfab property range from 2.7 ppb to 20 ppb, with the average being approximately 10 ppb. Soil PFOA concentrations follow a decreasing trend with distance from the former Chemfab facility (C.T. Male Associates, 2016).

No soil sample had a PFOA concentration approaching the Vermont Department of Health (VTDH) soil screening value (SSV) for exposure of 300 ppb. The results indicate that soils in the sampled areas do not pose a risk from direct soil exposure and that surface soil remediation is not necessary to address the soil PFOA concentrations encountered.

A liquid sludge sample collected at the Bennington WWTP by VTDEC had a PFOA concentration of 350 ppt.

VT ANR (2017) provided the following results.

- Pressed sludge sampling results: 5 samples, ranging in PFOA concentration from 6.9 to 8.2 ng/g, average 7.7 ng/g.
- Compost results: 1 compost sample, ND/<0.9 ng/g. SPLP sample 61 ng/l PFOA, 79 ng/l PFBS, 11 ng/l PFOS, ND/<4.0 PFHxS, 210 ng/l PFHpA, and 3 ng/l PFNA), and
- Septic tank sludge sampling (3 samples ranging from ND/<1.6 and <38 ug/kg to 69 ug/kg in one sample; this sample had 430 ng/l by SPLP).

### 4.2 Groundwater

PFOA sampling results for groundwater are shown on Figures 6 and 7. Figure 6 shows PFOA sampling locations and results for private wells (as of March 23, 2017) and the former Water Street Chemfab facility monitoring wells (as of December 2016). Figure 7 shows results of Bennington Landfill sampling. Within the study area, PFOA concentrations measured between March 2016 and March 2017 at private water supply wells and monitoring wells at the former Chemfab facility on Water Street and at Bennington Landfill range from non-detect to a high of 4,900 ppt.

#### 4.2.1 Private Wells

As of March 23, 2017, VTDEC had collected water samples for PFAS analysis at 606 private wells and water supplies (e.g., springs) within the study area. These wells are shown on Figure 6, and are symbolized with the most recent measured PFOA concentration in the VTDEC dataset. PFOA concentrations in private wells range from no detection in areas near the edge of the study area and between the former Chemfab facility on Water Street and Bennington Landfill up to 4,600 ppt near the former Chemfab facility on Water Street. (Note that the PFOA concentration of 4,600 ppt is from influent sampling for a residential point-of-entry-treatment (POET) system. The POET concentration dataset is not incorporated in the concentrations for private wells shown on Figure 6, which are based on the VTDEC dataset.) Less than half of the private wells sampled had concentrations exceeding the State's DWHA of 20 ppt. Approximately 37 percent had no detectable PFOA concentration, with a detection limit ranging from 2.0 to 6.7 ppt. Approximately 31 percent had PFOA concentrations exceeding 70 ppt.

Limited information on private well construction that would allow an evaluation of the vertical distribution of PFOA concentration is available to decipher concentration trends related to well construction. In general, wells tend to be completed in bedrock, but the available data suggest that some wells are completed in surficial materials. Also, those wells that are completed in bedrock have wide-ranging open interval depths, compared to wells completed in unconsolidated deposits.

#### 4.2.2 Monitoring Wells

Groundwater sampled in July and December of 2016 from monitoring wells at the former Chemfab facility on Water Street had PFOA concentrations ranging from 34 ppt to 4,900 ppt, with concentrations decreasing in the prevailing direction of groundwater flow near the facility (Figure 6).

A subset of Bennington Landfill monitoring wells were sampled by VTDEC and the samples analyzed for PFOA in March and April of 2016 (Weston Solutions, Inc., 2016). The wells sampled and PFOA concentrations are shown on Figure 7. Sample results show detections of PFOA of up to 140 ppt in the surficial monitoring wells, 5,300 ppt in the leachate vault, and non-detects (less than 2 ppt) in the two bedrock wells that were sampled. Some deep monitoring wells with historic detections of VOCs were not sampled. It was noted in the sampling report (Weston Solutions, Inc., 2016) that well B-4-3 could not be located at the time of sampling; the well has subsequently been located. VTDEC has confirmed that B-7-3 no longer exists, and that its former location is now a capped portion of the landfill. PFOS was detected in monitoring wells and private wells near the Bennington Landfill (see Section 7.0).

#### 4.2.3 Public Water Supplies

PFOA has not been detected in the municipal water supply for North Bennington, which was sampled in early February 2016, or the municipal water supply for Bennington (Town of Bennington, 2016), which was sampled in April 2016. The Bennington water supply also had no detectable PFOA (at detection levels of 3.3 ppt) during sampling in 2013 (Town of Bennington, 2016).

#### 4.3 Surface Water and Sediment

Surface water and sediment samples were collected by VTDEC in March 2016 at eleven locations at rivers, creeks, and ponds around the former Chemfab facilities in North Bennington and Bennington (State of Vermont, 2016). The locations are shown on Figure 8. (The location of the Paran Lake sample is shown on Figure 8 at an approximate location, as the VTDEC coordinates and map show it on the Walloomsac River rather than at Paran Lake.)

Concentrations of PFOA in surface water samples ranged from no detection to 79 ppt at the pond on the Bennington College campus. The pond on the Bennington College campus is a closed basin that receives overland runoff. The State of Vermont has indicated that "the PFOA concentrations found in the waters tested are much lower than concentrations that could be harmful to freshwater organisms, and are much lower than levels that would be a risk to people who swim there" (State of Vermont, 2016).

Concentrations of PFOA in sediment samples ranged from no detection to 2.4 ppb. The highest sediment concentration was found in the Walloomsac River below Paran Creek and has been stated by VTDEC to pose no risk to human health or aquatic species. A surface water sample taken at Bennington Landfill had a PFOA concentration of 34 ppt. A soil sample taken nearby had a PFOA concentration of 3 ppb (Figure 7).

## 5.0 Concept of PFOA Release and Transport

For purposes of this report, a working concept of PFOA release and transport for the study area is shown on Figure 9 and discussed below.

### 5.1 PFOA Release Points

The former Chemfab facilities in North Bennington and Bennington are among several locations within the study area where PFOA release to the environment could have occurred. The mechanisms for PFOA release include air emissions from industrial facilities and the transport, disposal, and leaching of PFOA-containing materials.

#### 5.1.1 Air Emissions from Chemfab Facilities

Air emission and deposition of PFOA as a source of regional groundwater PFOA detection has only recently been recognized. Most instances of groundwater detection by PFOA and PFOS have originated as one or more point sources (AFFF application, leaching from landfills, etc.). However, at some manufacturing facilities that make or use PF compounds, air emission and subsequent deposition on nearby land surfaces has been found to be a mechanism for PFOA to be transported to groundwater. The possibility for airborne emissions from industrial operations to use PFAS-containing materials to contribute PFOA to groundwater creates a high likelihood that there are numerous industrial sources of the PFOA detected in the study area.

For the purposes of this report, PFOA was considered to be emitted from the former Chemfab facilities as part of the fabric coating process. For purposes of this report, it is assumed that PFOA particulates were driven off the fabric during drying and released from the facility through the emission stacks.

Data regarding air emissions of PFOA during the period of facility operation are limited. For purposes of this report, air emissions of PFOA from the Bennington facility on Northside Drive were assumed to have begun in 1969 and ceased in 1978, when the facility was closed and operations were transferred to the North Bennington facility (on Water Street). For purposes of this report, the Water Street facility was assumed to have operated from 1978 until 2002, when the facility was closed. Based on operational data at a similar facility in Merrimack, New Hampshire, an estimate of PFOA content in the dispersions used in the coating process of 2,000 ppm (for high PFOA content dispersions) was applied to annual dispersions usage data and measured air emissions data to estimate annual PFOA emissions<sup>1</sup>. Many different dispersions were used at both facilities, and not all dispersions contained PFOA. Also, measured emissions varied based on operating process parameters. So, while any particular emissions scenario is not well known (i.e., knowing the emission rate for a specific stack on a specific day), when used at the level of this evaluation (to determine annual deposition estimates), the Merrimack facility data provide a conservative

<sup>1</sup> See Section 1.3.1 in Appendix A for a detailed discussion of air emissions estimates.

estimate of potential PFOA emissions from the Bennington and North Bennington facilities for purposes of this report.

The air deposition model was run using a five-year meteorological data set developed by VTDEC from the Bennington Airport. This data set is representative of meteorological conditions in Bennington, although these data are from a different time period (2006-2010) than when the facilities were in operation. Annual deposition was calculated for each year of facility operations. For purposes of this report, these emission estimates and modeling procedures are appropriate for the annual deposition rates which are input to the groundwater modeling analysis.

Emission estimates and modeling procedures are described in more detail in Section 6.1.1 and Appendix A.

### **5.1.2 Bennington Landfill Leachate**

Materials likely to contain PFOA were disposed of in Bennington Landfill. This is supported by high concentrations of PFOA in the landfill leachate measured in 2016 (5,300 ppt; Weston Solutions, Inc., 2016). PFOA and other PF compounds in leachate are likely derived from a mixture of commercial and residential (consumer) wastes that are known to contain PF compounds. An inventory and discussion of industries contributing waste to the landfill is provided in Section 2.4.4.

Historic data on PFOA concentrations of landfill leachate and groundwater near the landfill are not available. PFOA concentrations were not monitored during the site investigations and remedial activities beginning in the 1970s and continuing through the 2000s. Leachate was treated and infiltrated onsite from 1997 (TRC, 1998) to 2008 (VT ANR, 2016). PFOA concentrations in the leachate and the effectiveness of the treatment system in removing PFOA during this time are not known since sampling of treatment system effluent for PFAS did not occur. Leachate is currently not being treated onsite; since 2008, the vault is reportedly pumped out approximately every other year and disposed offsite (VT ANR, 2016). Without treatment of leachate, the landfill has the potential of being a continuing source of PFOA to the groundwater system. Groundwater flow in bedrock from the landfill is towards the south and southwest, in the direction of private water supply wells where PFOA has been detected. Evaluation of groundwater flow conditions and groundwater quality in and near the Bennington Landfill is warranted to determine if the landfill is causing exceedances of regulatory standards for PFOA downgradient of the landfill. Because PFOA does not degrade (unlike many landfill-related constituents), is poorly adsorbed (unlike other landfill-related constituents), and requires detection limits less than 10 ppt (compared to standard method detection limits for volatile organic compounds and other landfill-related constituents that are 100 to 1,000 times higher), careful sampling and specialized analytical procedures may be necessary to draw correlations between conventional landfill constituents and PFOA from the landfill.

Dewatered municipal wastewater treatment sludge from the Bennington WWTP was reported to be temporarily stored in areas abutting the site (these areas are identified in Appendix I of the August 1992 Site Investigation Work Plan). Tarps were placed over the stockpiles to prevent rain infiltration. The sludge was placed onsite to be used as part of the capping material pursuant to CERCLA closure (McLaren/Hart, 1997).

### 5.1.3 Land Application of WWTP Sludge

Disposal of sewage sludge produced during treatment of wastewater at the Bennington wastewater treatment plant (Figure 1) may have occurred through land application at various locations in the surrounding area. Limited data exist regarding the WWTP sludge disposal locations and volumes. Bennington Landfill is the only documented disposal location (Section 2.4.3), although anecdotal information supplied by VTDEC indicates that sludge from the Bennington WWTP may have been supplied to local property owners for application to agricultural lands in the area before such practices were prohibited by the State of Vermont. No data exist on PFOA concentration in the sludge at the time of disposal, however, research indicates that PFAS are commonly found in wastewater treatment sludges (Zareitalabad et al., 2013). As a result, the specific PFOA mass contributed to the groundwater system through this release mechanism is unknown, but it is likely that application of WWTP sludge has contributed to detections of PFOA in soil and groundwater in the study area.

### 5.1.4 Disposal at the Former Chemfab Facilities

Potential releases at the former Chemfab facilities in addition to air emissions could theoretically include disposal of liquid wastes or residues. However, to date, there is no indication that disposal of PFOA and release to the environment occurred in this manner to any significant degree. Investigation activities at the facilities are ongoing.

### 5.1.5 Other Landfill Areas

Now-closed landfills were previously in use on the Bennington College property that could be sources of PFOA. There is also a likely landfill associated with a former paper plant on North Bennington Road at Murphy Road. Because landfills are known sources of PFOA, PFOS, and other PF compounds from commercial and consumer materials, these disposal sites may be sources of PF compounds in the groundwater in the area. VTDEC has also indicated that PFOA has been detected in monitoring wells at the Kocher Drive Landfill.

### 5.1.6 Other Industrial and Commercial Users

Many of the waste disposers for Bennington Landfill are also potential sources of PFOA because they likely used PFAS in their processes. These waste disposers are discussed in Section 2.4.4.

There are numerous other potential sources of PFAS from a wide variety of industrial and commercial activities in the study area, including but not limited to car washes; granite, stone and tile finishing; storage and/or use of fire-fighting foam; wood floor finishing; automotive repair; junkyards; incinerators; carpet and upholstery cleaning; and painting/coating application.

## 5.2 PFOA Transport Pathways and Mechanisms

For purposes of this report, the PFOA transport pathways and mechanisms for the study area were assumed to include different media (air, soil, water) and different levels of subsurface water saturation (partially and fully saturated). The transport pathways connect source to receptor (e.g., private well) with various potential storage mechanisms along the way (e.g., soil retention). Figure 9 shows the conceptual PFOA release and migration processes and pathways.

### 5.2.1 Air Transport and Deposition

For purposes of this report, particulates of PFOA released in stack emissions were represented as being transported by wind and deposited at the ground surface. The pattern and magnitude of air deposition are dependent on the PFOA emission rate, the relative size distribution of PFOA particles, the wind conditions at the time of release, and the factors that would influence air transport away from the facility (e.g., topography). Larger particles settle out of the air column and are deposited at the ground surface closer to the facility, compared to smaller particles, which are deposited farther from the emission source. More PFOA particles are deposited closer to the facility during low wind conditions. As a result, the PFOA mass deposition rate generally decreases with distance from the facility. PFOA deposition onto roofs and other impervious surfaces, along with precipitation runoff, could constitute a potential local source of PFOA at the storm water outfall.

### 5.2.2 Dissolution in Water and Infiltration

For purposes of this report, PFOA deposited at the ground surface is transported into the subsurface by infiltrating precipitation. The concentration of PFOA in the infiltrating water is a function of the PFOA mass deposition rate, the ability of the soils to retain PFOA, and the rate of infiltration.

For the purpose of this report, the rates are assumed to be constant throughout a single year, so the annual rates (inches per year, in/year; pounds per year, lb/year) can be considered as annual volumes.

### 5.2.3 Unsaturated Zone and Groundwater Transport

Once PFOA infiltrates below the ground surface, it leaches across and is temporarily retained in the unsaturated zone – the subsurface area above the water table – before reaching the groundwater system.

The rate of PFOA transport in the subsurface (both the unsaturated zone and groundwater system) has been shown to be controlled primarily by adsorption onto organic matter in the subsurface material, specifically organic carbon (Zareitalabad et al., 2013). PFOA sequestration can also result due to sorption onto neutrally-charged, high-surface-area particles (e.g., some silts and clays) or electrostatic interactions with positively-charged surfaces (e.g., iron hydroxide coatings on soil surfaces). However, these attenuation mechanisms are not significant in comparison to adsorption onto organic carbon (Higgins and Luthy, 2006). Thus, a characterization of PFOA retention by adsorption to organic carbon as defined by  $K_{oc}$  coefficients is slightly low, because the minor influences of iron oxides and mineral phases are not explicitly considered.

The sorption of PFOA to subsurface materials acts primarily to slow its transport relative to the rate of water movement. Due to the non-reactive nature of PFOA, degradation is limited and does not account for significant reduction of total mass in the system.

While total mass is conserved, the slowing of transport across the unsaturated zone can have important implications for mass loading to the groundwater system by reducing the peak loading value and spreading out loading over a longer period of time (i.e., dampening and lagging the concentration response to a release of finite duration).

As a practical matter, once PFOA migrates below the root zone, the opportunity for uptake by plants ends. Over time, PFOA that is below the root zone will migrate to the water table. The time required to reach the water table may be several years, depending on the moisture conditions, the amount of precipitation, the depth to the water table, the amount of organic matter in the unsaturated zone, and the saturated permeability of the unsaturated zone materials.

#### 5.2.4 Conditions at Bennington Landfill

The hydrogeologic setting of Bennington Landfill differs from the conceptual model for the region in that groundwater in the unconsolidated deposits at and near the landfill is perched above the groundwater in the bedrock. In other words, the base of the shallow, saturated unconsolidated materials overlies unsaturated bedrock and weathered bedrock (saprolite). This is an indication that the rate of leakage through the bottom of the saturated unconsolidated materials is not sufficient to cause complete saturation of the underlying bedrock and weathered bedrock.

PFAS have been detected at concentrations exceeding regulatory standards in landfill monitoring wells. The leachate collection vault had a PFOA concentration of 5,300 ppt when sampled by EPA in spring 2016 (Figure 7). The leachate is pumped from the vault periodically for offsite treatment and disposal. In addition to PFOA, PFOS has been detected in landfill monitoring wells and the PFOA+PFOS concentrations exceed 20 ppt in most of the sampled wells. Measured PFOS concentrations are approximately one to two orders of magnitude lower than the corresponding PFOA concentrations (Figure 7).

This situation presents the possibility that some constituents reaching the aquifer in the unconsolidated aquifer may be discharged from the shallow groundwater flow system without reaching the underlying bedrock aquifer. Sampling of two existing bedrock wells by VTDEC did not detect PFOA (Weston Solutions, Inc., 2016). A third bedrock well (B-4-3) was located in 2017; this well will be redeveloped and sampled as part of an upcoming investigation. The fourth bedrock well (B-7-3) is understood to no longer exist. Historical water-quality data indicate that leakage from the unconsolidated aquifer to the bedrock aquifer did take place at rates sufficient to cause detectable concentrations of landfill-sourced constituents in bedrock wells, such as VOCs. Because VOCs both adsorb and degrade over time, their concentrations will be attenuated with respect to PF compounds, which neither degrade nor sorb to any meaningful extent in the absence of organic carbon.

## 6.0 Modeling of PFOA Fate and Transport

The objectives for the modeling described below are presented in Section 2.1. The complete PFOA transport pathway from source to potential receptors was simulated. As indicated above, these transport pathways are through air, unsaturated soil and/or bedrock, and saturated aquifer materials.

No single model in standard practice exists that can account for transport through these different media. Therefore, transport was simulated by a series of distinct models, with the output of one or more models serving as input for the model simulating the next segment of the transport pathway. Each of these models is a widely accepted and well-documented tool for the transport pathway to which it was applied.

### 6.1 Modeling Approach, Methods, and Assumptions

The approach to modeling PFOA fate and transport for purposes of this report consisted of combining distinct models of the various components of the transport pathway: air, unsaturated zone, and groundwater. The flow chart on Figure 10 shows the relationship and linkages between the various models and the position of each model in the overall transport pathway. In this way, the primary physical processes that control PFOA fate and migration are included in the analysis.

Depending on the position of each segment of the overall transport path, the model representing the segment may provide output to other models while also requiring the results of another model as input. For example, the air model does not receive model output as an input, but its results are passed as input to the unsaturated zone model.

Each of the pieces of the modeling approach are discussed in turn in the following subsections. The assumptions involved in the overall modeling approach are discussed in Section 6.1.5. Full details of each modeling component can be found in the following appendices:

- Appendix A - Air Dispersion and Deposition Modeling
- Appendix B - Estimation of Infiltration Rates Using the Soil-Water Balance Model
- Appendix C - Simulation of Unsaturated Zone Flow and Transport
- Appendix D - Groundwater Flow and Solute Transport Modeling

#### 6.1.1 Air Dispersion and Deposition

For purposes of this report, emission of PFOA from stacks at the former Chemfab facilities and the resulting air dispersion and deposition were simulated using AERMOD (USEPA, 2016), which is the U.S. Environmental Protection Agency's preferred dispersion model for regulatory modeling analyses.

AERMOD has been demonstrated to be an effective tool for simulating air dispersion and deposition of PFOA and PFOS (Barton et al, 2010). Limited information is available regarding actual PFOA emissions during operations from 1969 – 2002. Modeled emissions were estimated from annual dispersions usage and this analysis is adequate for the purposes of this report in estimating PFOA deposition over a long period of time.

The air dispersion and deposition modeling analysis incorporated emissions data with hourly meteorological data and topography to estimate annual PFOA ground surface deposition at each model node. For a given emission scenario and time series of meteorological conditions, the model calculates PFOA mass transport through the air and ground surface deposition at each of the model nodes.

A single air dispersion model was developed to include both the North Bennington (Water Street) and Bennington (Northside Drive) facilities. The air dispersion model grid consists of 8,394 nodes over a 14 km x 13 km area approximately centered on the facilities (Figure 11). Model nodes are spaced at 20-meter intervals in the areas around the Water Street and Northside Drive facilities. Nodes are spaced at 200-meter intervals outside of the areas of refinement. The 20-meter spacing aligns the air model nodes with the cells of the unsaturated zone and groundwater models, which allows passing of the simulated PFOA mass deposition values directly (i.e., with no interpolation) to the unsaturated zone model.

The air dispersion model was run separately for the Northside Drive facility and the Water Street facility, although each facility was modeled using the same five-year (2006 - 2010) meteorological data set from Bennington Airport. Use of a five-year meteorological data set is recommended by USEPA<sup>2</sup> to adequately capture inter-annual meteorological variability.

Operations were transferred from the Northside Drive facility to the Water Street facility in mid-1978. For this analysis, as a simplifying assumption, all of the emissions in 1978 were assumed to be emitted from the Water Street facility. This simplifying assumption includes all of the estimated air emissions in the model, but gives a very slight change to the temporal and spatial component of the emissions. The Northside Drive facility emissions for each year were assumed to be continuously emitted from a single representative stack. Annual emission estimates for the Northside Drive facility ranged from 13 lb (1970) to 104 lb (1975) with an average of 47 lb/year.

The Water Street facility was modeled for 1978 – 2001 when the facility was in operation. The facility closed in February 2002, so 2002 was excluded from the modeling analysis. The air emissions from the Water Street facility were assumed to be emitted primarily from the process stacks. However, based on facility inspections by VTDEC personnel, visible emissions were seen from the cupola vents, which may potentially have contained PFOA. To account for this theoretical possibility, 5% of total emissions at the Water Street facility were assumed to be fugitive emissions without abators and were modeled accordingly. Annual emission estimates for the Water Street facility ranged from 26 lb (1978) to 307 lb (1998) with an average of 145 lb/year.

Additional modeling options used in the analysis (and described in detail in Appendix A) include:

- Particulate settling
- Terrain elevations
- Stack parameter data

<sup>2</sup> Appendix W of 40 CFR Part 51 – Section 8.3.1.1

- Building downwash effects (caused by turbulent air flow downwind from an obstruction)

Where necessary, the simulated mass deposition rates for each emissions scenario were interpolated to unsaturated zone model nodes using ordinary kriging, as encoded in the Surfer software program (Golden Software, Inc., 2011).

### 6.1.2 Estimation of Infiltration Rates

For purposes of this report, the Soil-Water Balance (SWB) model (Westenbroek et. al., 2010) was used to calculate spatial and temporal variations of infiltration rates across the study area. Details of the SWB model development are included in Appendix B.

The SWB model calculates components of the soil water balance on a daily basis. The SWB model domain is shown on Figure 12. Inputs to the SWB model include the following:

- Daily precipitation and daily minimum and maximum temperatures: These data were obtained for a single location representing the North Bennington area. Data were obtained for the period 1980 to 2015.
- Land cover classifications: These data were obtained from the National Land Cover Database.
- Hydrologic soil group and soil water capacity: These data were obtained from the Natural Resources Conservation Service soil survey geographic database.
- Surface flow direction as inferred from topography: Topography data were obtained from the U.S. Geological Survey seamless elevation dataset.

Monthly infiltration grids, as output from the SWB model, were used to define the 1980-2015 average infiltration rate on a cell-by-cell basis and to develop a mass flux concentration for the solute transport model. The recharge boundary in the groundwater flow model was defined by the gridded 1980-2015 average infiltration rates. The 1980-2015 average PFOA concentrations in the infiltrated water were calculated by dividing the average simulated PFOA mass deposition rate from the air deposition model by the 1980-2015 average simulated infiltration rate from the SWB model; these values were used as input to the unsaturated zone model.

### 6.1.3 Unsaturated Zone Modeling

For purposes of this report, leaching and retention of PFOA in the unsaturated zone was simulated to provide a physical link between the infiltration rates from the SWB model, the mass deposition rates from the air model, and the inputs for the groundwater transport model described below. The recent detections in shallow soil suggest that unsaturated zone retention is an important process in PFOA fate and transport and that leaching has not completely flushed PFOA from the shallow soils in the years since emissions ceased at the former Chemfab facilities.

For purposes of this report, leaching and retention of PFOA in the unsaturated zone was simulated using the MODFLOW-NWT (Niswonger et al., 2011) and MT3D-USGS (Bedekar et al., 2016a; 2016b) codes, specifically the unsaturated zone flow (UZF) and unsaturated zone transport (UZT) packages of the codes.

The unsaturated zone modeling methods and assumptions made in applying those methods are discussed below and in detail in Appendix C.

For purposes of this report, the unsaturated zone processes were simulated for a series of vertically-oriented, one-dimensional profiles representing the range of conditions in the study area. The base model configuration and parameter values for the unsaturated zone model are shown on Figure 13.

The vertical profile models were developed to account for the variability in unsaturated zone properties across the site that would affect PFOA leaching: infiltration rate, PFOA mass deposition rate, and simulated unsaturated zone thickness. In order to preserve a computationally manageable approach, the distribution of values for each of these properties was split into a number of representative bins, and a profile model was developed and run for each unique combination of property value bins. A schematic example of this process is shown on Figure 13. For purposes of this report, the simulated PFOA concentrations at the water table over the simulation period (1969-2052) were then mapped back to each of the model cells in the uppermost layer of the groundwater model through the combination of the values at the cell location.

For purposes of this report, water flow through the unsaturated zone was simulated as a steady-state (constant) rate equal to the average annual value of simulated infiltration from the SWB model. Preliminary simulations in which the flow rate changed annually did not show variation in PFOA concentration at the depth of the water table.

Material types were consistent between simulated profiles and selected from generic soil types in the HYDRUS-1D database (Šimůnek et al., 2009). This database was used to provide representative parameter values for the mathematical function describing water movement in the unsaturated zone.

For purposes of this report, sorption of PFOA was simulated with a linear isotherm at equilibrium conditions. No degradation was simulated. With this sorption isotherm, the water concentration is linearly related to the solid-phase (sorbed) concentration using a distribution coefficient ( $K_d$ ).  $K_d$  is the product of the fraction of organic carbon ( $f_{oc}$ ) and the organic carbon partition coefficient ( $K_{oc}$ ). In order to obtain  $K_d$  values, representative values of  $f_{oc}$  and  $K_{oc}$  were assumed. Values of  $f_{oc}$  were assumed for each material type: 0.025 for clay loam, 0.005 for sandy clay loam, and 0.0005 for sand, consistent with published values (Zheng and Bennett, 2002) and total organic carbon values measured in shallow soil samples collected at and near the former Water Street Chemfab facility (C.T. Male Associates, 2016). For all material types, a  $K_{oc}$  value of 550 cubic centimeters per gram ( $\text{cm}^3/\text{g}$ ) was arrived at through the transport model calibration process, which is consistent with published values showing an average laboratory-based  $K_{oc}$  value for PFOA of 630  $\text{cm}^3/\text{g}$  ( $\log K_{oc}$  ( $\text{cm}^3/\text{g}$ ) = 2.8; Table 1 of Zareitalabad et al., 2013).

The sorption process with the linear isotherm is reversible, meaning that as PFOA concentrations in the infiltrating pore water increase, more PFOA is adsorbed to the organic fraction of the soil. As concentrations in the pore water decrease, PFOA is desorbed back into the infiltrating pore water. As PFOA concentrations in pore water approach zero, the final adsorbed mass is desorbed and becomes a

dissolved concentration. All PFOA mass is conserved in this processes – it all eventually makes its way through the unsaturated zone to the water table.

For purposes of this report, the unsaturated zone thickness was varied to account for the variability across the study area simulated by the groundwater flow model (described in the next section). Thickness of the deepest material type in the model (sand) was adjusted to account for the changing thickness of the unsaturated zone.

#### 6.1.4 Groundwater Flow and Solute Transport Modeling

For purposes of this report, groundwater flow was simulated using the MODFLOW-NWT code (Niswonger et al., 2011). PFOA transport was simulated using the MT3D-USGS code (Bedekar et al., 2016a, 2016b). The modeling methods and assumptions made in applying those methods are discussed below and in detail in Appendix D.

The domain of the groundwater flow model extends to the outer boundaries of the Hydrologic Unit Code (HUC) 12 watersheds surrounding the HUC 12 watershed in which the former Chemfab facility is located. The model domain covers approximately 140 square miles and is shown on Figure 14.

An irregular grid was used with 20 x 20 meter cells near the former Chemfab facility in North Bennington and cells ranging up to approximately 230 x 230 meters in the far field. The MODFLOW model has two layers, the upper layer representing the surficial material, where present, and the lower layer representing bedrock. (At bedrock outcrops, both model layers represent bedrock.) The unconsolidated deposits were combined into zones with the same predominant lithology for the purpose of modeling, as shown on Figure 2A. (Note that the interpretation on Figure 2B from DeSimone (2017) was recently completed and was not determined to be sufficiently different from previous interpretations to warrant modification of the modeling work described in this report.) The mapped bedrock units shown on Figure 3 were grouped by material type (coloring on Figure 3) and these groupings were used to define zones in the bedrock model layer. One extra zone was added to both model layers near Bennington Landfill to allow simulation of the perched groundwater noted at that location.

For purposes of this report, the recharge package of MODFLOW-NWT was used to simulate infiltrating water that reaches the water table. This was the only source of water simulated in the model domain. Values of recharge were derived from the SWB model described in Section 6.1.2. The drain package was used to simulate streams (brooks) and rivers in the model domain.

Calibration of the simulated steady-state groundwater flow field to head data as measured in wells and an estimated baseflow value for the Walloomsac River was accomplished using the parameter estimation code PEST (Watermark Numerical Computing, 2005; 2015). Results of the calibration are provided in Appendix D.

For purposes of this report, PFOA transport through the groundwater system was simulated using the MT3D-USGS code (Bedekar et al., 2016a; 2016b) and a steady-state groundwater flow field simulated by MODFLOW. MT3D-USGS is linked to MODFLOW and uses the flow field and budget information

produced by MODFLOW as the advective (flow-driven) component of transport. PFOA mass was input to MT3D-USGS by assigning a recharge rate determined using the SWB model and the time series of PFOA concentration at the water table estimated by the unsaturated zone model, as described above.

Transport property values used in MT3D-USGS included effective porosity values of 0.3 for cells representing unconsolidated deposits and 0.07 for cells representing bedrock, a longitudinal dispersivity value of 10 meters, a horizontal transverse dispersivity to longitudinal dispersivity ratio of 0.1, and a vertical transverse dispersivity to longitudinal dispersivity ratio of 0.01. Sorption, while simulated using the same parameters as in the unsaturated zone, was of negligible importance due to the low values of organic carbon. Diffusion, and decay in the groundwater system were not simulated based on the observed transport properties of PFOA. PFOA adsorbs poorly to the aquifer materials (Appendix D, Section 2.3) and does not degrade compared to commonly encountered groundwater constituents.

### 6.1.5 Assumptions

Assumptions made in applying the various models used in the modeling framework are listed below.

For purposes of this report, the following assumptions were made in applying AERMOD to simulate air dispersion and deposition:

- PFOA emissions were simulated at a continuous rate throughout each model year (i.e., emissions were modeled for each hour of the year).<sup>3</sup>
- Emission rates were assumed to be proportional to annual dispersions usage.
- Emissions were assigned to stacks in operation during the year modeled.
- Emissions were apportioned based on capacity (mmbtu) of the abator at each tower.
- The Bennington Airport 2006 – 2010 meteorological data set is representative for the Bennington and North Bennington areas, even though the meteorological data set time period is not concurrent with the facility emissions.

For purposes of this report, the following assumptions were made in applying SWB and MODFLOW to simulate groundwater flow:

- Variation in precipitation and daily minimum and maximum temperature across the SWB model domain are adequately represented by data compiled for a single location.
- Assumptions inherent in the use of SWB. The SWB results compare favorably with published estimates of infiltration in the region (see Appendix B).
- Runoff leaving the model domain does not remove a significant amount of PFOA mass. Note that simulation of runoff from one part of the model domain to another does not remove PFOA from the model domain.

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<sup>3</sup> The Water Street facility typically operated 24 hours/day, 5 days/week. While AERMOD has the ability to accommodate a variable schedule, this level of model refinement is not relevant here considering that annual emissions are important (and not short-term variability) and the model does not use concurrent meteorological data.

- Groundwater does not flow across the watershed divides used to define the model domain. The basis for this assumption is presented in Section 3.7.
- Pumping from wells in the model domain is not known to alter the regional flow paths established between the recharge at the water table and discharge to surface water features. Pumping from residential wells is low and diffuse. High capacity commercial and industrial wells were not found to be present in the area. Therefore, pumping of the wells was not simulated in the MODFLOW model.
- The surficial materials and bedrock are assumed to be equivalent porous media (EPM) at the simulated scale. At a sufficiently large scale, an interconnected network of secondary porosity features (e.g., fractures) can be represented as an equivalent porous medium and groundwater flow can be modeled using standard methods for materials in which flow is through primary porosity features (Anderson, Woessner, and Hunt, 2015).
- All streams simulated with the drain package are assumed to be gaining. If significant pumping of groundwater occurs from a well or group of wells located near a surface water body simulated by the drain package, any recharge from the surface water body to the groundwater system is not accounted for in the model.

For purposes of this report, the following assumptions were made in applying MT3D-USGS to simulate PFOA transport in the unsaturated zone:

- Simulation of infiltration at the long-term average rate estimated using SWB adequately captures the effect of infiltration rates that vary throughout each year and from year to year.
- The distribution of organic carbon used in the model, which has the same variation with depth in each model cell, adequately simulates the spatial variation of this organic carbon in the model domain.

For purposes of this report, the following assumptions were made in applying MT3D-USGS to simulate PFOA transport in the saturated groundwater flow system:

- As noted above, infiltration from the ground surface is simulated at steady-state rates. In effect, this involves an assumption that the time lag between infiltration of water past the root zone and recharge at the water table is insignificant compared with the time lag associated with sorption and desorption of PFOA in the shallow soils containing organic carbon.
- PFOA does not adsorb to the materials that make up the aquifer system in the saturated model domain (i.e., organic carbon in the saturated materials is negligible).
- PFOA does not degrade under the conditions simulated.
- The aquifer in the unconsolidated material and the bedrock aquifer can be adequately represented at the regional scale of the model using one model layer for each unit.
- Dual-porosity (aka matrix-diffusion) processes are not explicitly simulated, however the modeling methods used for the aquifer system implicitly simulate the effects of these processes (See discussion in Section 6.2.7.2 and Appendix D, Section 2.5).

The following processes are not accounted for in the modeling approach used here:

- Sources of PFOA other than the former Chemfab facilities, such as land application of biosolids, releases from businesses or households using PFOA-containing materials, or landfilling of PFOA-containing materials.
- Uptake in plants that have not entirely degraded and released PFOA back to the environment.
- Cut-and-fill operations that remove soils that contain PFOA.
- Cut-and-fill operations that introduce soils that contain PFOA.

## 6.2 Modeling Results and Discussion

### 6.2.1 Air Deposition Values and Patterns

Plots of PFOA mass deposition rates simulated by the air model for each year in the period 1969-2001 are included in Appendix A. The average simulated air deposition rates for the period 1969-2001 are shown on Figure 15 for the simulated air emissions scenario.

The simulated mass deposition rates are greatest near the former Water Street Chemfab facility, which is consistent with the air transport and deposition concepts discussed in Section 5.2.1. There is some variability around the facility due to the variability in meteorological conditions. Deposition around the former Northside Drive Chemfab facility is relatively low, given the relatively low average emissions rate of 47 lb/year. Both emission sources are considered simultaneously in the modeling as part of this evaluation.

### 6.2.2 Infiltration Rates and Concentrations

For purposes of this report, monthly infiltration grids, as output from the SWB model, were used to define the recharge boundary in MODFLOW and develop a mass flux concentration for the groundwater transport model (see Section 3.6 for details). The average annual infiltration for 1980-2015 is shown on Figure 16. Plots of annual infiltration for each year simulated (1980-2015) are included in Appendix B. Across the model domain, the average infiltration during this period was 19.5 in/year, averaging 43 percent of annual precipitation. These results are generally consistent with estimates of recharge based on streamflow records. Jerris and DeSimone (1992) provided a range of groundwater recharge values of 20 to 35 in/year, depending on elevation based on differences between precipitation records and stream gaging records for an area south of the SWB model domain. Flynn and Tasker (2004) estimated groundwater recharge to range between 17.9 and 28.1 in/year for four different stream basins located east and northeast of the SWB model domain.

For purposes of this report, concentrations of PFOA in water prior to infiltration (pre-infiltration) were estimated for each year in which air deposition was simulated by dividing the PFOA mass deposition rate simulated by the air model by the average simulated infiltration rate for 1980-2015 from the SWB model. It is necessary to calculate the pre-infiltration PFOA concentration because it is applied at the upper boundary condition of the unsaturated zone model. An example of the pre-infiltration concentration magnitudes and distribution is shown on Figure 17. The concentrations for 2001, the last year with air emissions and deposition, range from less than 50 ppt to greater than 7,000 ppt near the former Water Street Chemfab facility.

### 6.2.3 Effect of Unsaturated Zone Transport on PFOA Breakthrough

The delayed transport through the unsaturated zone, related to both the time lag associated with water flow through the unsaturated zone and the retention of PFOA due to sorption processes, results in PFOA breakthrough at the water table that is muted and delayed compared to concentrations that would result if transport across the unsaturated zone were assumed to be instantaneous. This effect is shown on Figure 18 for an evaluation point east of Paran Creek from the former Water Street Chemfab facility: simulated soil concentration peaks at approximately 30 ppb in 1998, but PFOA concentration in unsaturated zone water just above the water table does not peak until 2006.

When considering the timing and magnitude of peak PFOA concentration in the groundwater system, the temporary retention of PFOA in the unsaturated zone is important as the peak groundwater concentrations may occur after air deposition has ceased. (This is the case for the example location shown on Figure 18.) Graphs of simulated concentrations through time at selected well locations are presented in Appendix D.

### 6.2.4 Comparison to Soil PFOA Concentrations

The results of the unsaturated zone modeling are comparable to measured soil concentrations. This comparison was made to evaluate which  $K_{oc}$  value would provide simulated soil concentration values that reasonably match measured concentrations. Given the uncertainty in the parameter values used to account for PFOA sorption in the subsurface, these comparisons are qualitative. Figure 18 shows the simulated and measured soil concentrations at the comparison location east of the former Water Street Chemfab facility, when using the  $K_{oc}$  value of 550 cm<sup>3</sup>/g that was arrived at through model calibration and a  $f_{oc}$  value for the upper 10 cm of soil of 0.025. These values are very close to the literature value for  $K_{oc}$  (630 cm<sup>3</sup>/g) discussed in Section 6.1.3 and the measured total organic carbon for the soil sample (22,300 mg/kg, i.e.,  $f_{oc} = 0.0223$ ; C.T. Male Associates, 2016).

### 6.2.5 Comparison to Groundwater PFOA Concentrations

Simulated PFOA concentrations in groundwater are included in Appendix D for each year in the simulation period. Figure 19 shows simulated PFOA concentrations in groundwater in 2016 for the simulated air emissions scenario, which allows a comparison of simulated and measured values. There is more variability in concentrations for the unconsolidated sediments given the variability in saturated thickness in the model. Figure 20 shows simulated and measured PFOA concentrations over time at one well location. Additional plots are included in Appendix D.

Not all concentrations observed in the field can be reproduced by a groundwater flow and solute transport model, especially where there are other potential sources (Konikow, 2011). For these reasons, a detailed comparison of model results to site data is not warranted for this report. Rather, a general comparison between areas simulated as being affected and those shown as not being affected from the simulated source is a reasonable use of model results in this context.

Factors that would affect the agreement between available data and model results include: